Islands in cities: urbanization and fragmentation drive taxonomic and functional variation in ground arthropods

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Abstract

The conversion of natural lands in urban areas is exponentially increasing worldwide, causing a major decline in biodiversity. Environmental alterations caused by urbanization, such as land conversion and isolation of natural patches, favour tolerant and generalist species, causing both species loss and replacement. In addition, selective pressure is exerted on particular functional traits, driving a functional homogenization or turnover of biotic communities. We sampled ground arthropods within the municipality of Turin (NW-Italy), wherein an isolated and a connected control subplot were repeatedly sampled at 15 stations distributed along a gradient of increasing urbanization. Such a nested sampling design allowed us to investigate the taxonomic and the functional responses of carabids and spiders to both the urbanization level and patch isolation. First, we highlighted the dominant role played by species homogenization (*nestedness*) in explaining both taxonomic and functional variation in both groups of arthropods. Secondly, we showed that urbanization causes simultaneously functional homogenization and replacement in both carabid and spider assemblages, whereas patch isolation influences carabid species composition and homogenizes and shifts spider taxonomic and functional composition. Lastly, by relating community-weighted means of body length, dispersal capacity and trophic strategy to the urbanization and isolation gradients, we demonstrated that urbanization alters the trophic structure of both taxonomic groups and increases the average dispersal capacity of spiders. On the other hand, patch isolation affected the

functional composition of spiders only, reducing the body size and increasing dispersal capacity and the proportion of web-builder species. Our results demonstrate that both urbanization and patch isolation alter species composition by causing functional and taxonomic homogenization. In addition, they exert a strong filtering effect on community functional traits, increasing the proportion of phytophagous species in carabids, and increasing dispersal capacity and web-builders occurrence in spiders, while reducing spider body size.

Keywords: taxonomic homogenization; functional homogenization; carabids; spiders; traits

Introduction

Throughout recent human history, the conversion of natural lands in urban areas increased exponentially everywhere in the world (Seto et al., 2011), representing today one of the main anthropogenic impacts on natural ecosystems (Grimm et al., 2006). In particular, the process of urbanization induces major detrimental effects on the environment, as a result of the severe and fast changes in landscape structure and physical conditions (Parris, 2016). These alterations are common to all urban settlements, and they are expected to drastically affect biodiversity in all continents (McKinney, 2006). However, although an increasing number of studies investigated the relationship between urbanization and species richness, the underlying mechanisms that drive biodiversity patterns in urban areas are not fully understood (but see Shochat et al., 2006, 2010; Pickett et al., 2011; Turrini & Knop, 2015), with inconsistent results among the studied taxa (see Chace & Walsh, 2006; McKinney, 2008; Niemela & Kotze, 2009; Luck & Smallbone, 2010; Jones & Leather, 2012; Ramirez-Restrepo & MacGregor-Fors, 2017, Piano et al., 2019).

From an evolutionary point of view, urbanization represents a primary selective pressure on biotic communities, especially when considering the hindrance to the establishment and/or survival of certain species (McKinney, 2002). Such a strong environmental filter is expected to differentially affect species from the regional pool, by either favouring tolerant taxa or disfavouring the most sensitive ones, with subsequent changes in species composition, mostly due to species loss (Sadler et al., 2006; Magura et al., 2008; Niemelä & Kotze, 2009; Piano et al., 2017) or to species replacement (Sattler et al., 2010; Vergnes et al., 2014; Knop, 2016) along the urbanization gradients. As a result, both phenomena result in biotic homogenization, due to the loss of rare and specialised species, and the gain of widespread tolerant ones (McKinney, 2006).

In this context, a prerequisite to properly describe the mechanisms that drive variations in taxonomic and functional community composition (β-diversity) is the evaluation of the relative contribution of *turnover* and *nestedness*, whereby *turnover* measures the degree to which species are replaced by others—species replacement—and *nestedness* measures the degree to which communities of species-poor sites are a subset of those in species-rich sites—i.e. species loss (Ulrich, Almeida-Neto & Gotelli, 2009). To our

59 knowledge, this approach has never been applied to investigate ground arthropod communities in urban
60 areas (but see Brice, Pellerin & Poulin, 2017 and Gianuca et al., 2017 as examples in other contexts).
61 According to the "habitat templet theory" (Southwood, 1977, 1988), the above-mentioned selection
62 process may have major effects on particular functional traits, like morphological or physiological
63 attributes, or it could affect the realized niche of a certain species (Webb et al., 2010). Interestingly, several
64 studies confirmed this hypothesis, providing evidence that, by filtering species according to their life-history
65 traits and ecological needs, urbanization often determines shifts in life-history community traits, i.e.
66 functional replacement (e.g. Vergnes et al., 2014; Concepción et al., 2015; Gianuca et al., 2017; Merckx et
67 al., 2018), or deletions of some functional groups, i.e. functional loss (e.g. La Sorte et al., 2014; Piano et al.,
68 2017). As a consequence, urbanization may favour species with common traits conferring positive fitness
69 under local conditions, causing the overall functional homogenization of urban communities (Olden et al.,
70 2004).
71 The study of urban-rural gradients has been successfully used to highlight the effects of urbanization on
72 biotic communities (McDonnell & Hahs, 2008). However, urbanization gradients are indirect and complex,
73 since they include different types of disturbance acting at different spatial scales (Pickett et al., 2011; Parris,
74 2016), and changes in community may depend on several factors associated with urbanization (Rebele,
75 1994; Seto et al., 2011). For instance, urbanization correlates with a higher presence of impervious
76 surfaces, which are known to have an effect on temperature, soil nutrient cycling and gas exchange and
77 other physical or chemical parameters (Parris, 2016). In addition, the growth of urban areas is responsible
78 for natural and semi-natural habitat fragmentation, which creates small and isolated residual patches,
79 which are typically highly dynamic and impermanent ecosystems (Parris, 2016). Accordingly, since physical
80 and landscape alterations are strongly correlated, even acting synergistically along urbanization gradients
81 with similar demographic consequences, it is often difficult to distinguish their effects on biotic
82 communities. Therefore, in order to isolate and describe their effects on biodiversity, a hierarchical
83 approach is required, taking into consideration the nested nature of ecological systems (McDonnell & Hahs,
84 2008).

In this research, we aimed to assess which mechanisms are involved in shaping ground arthropod assemblage composition in urban areas, focusing on the role of taxonomic/functional *turnover* and/or *nestedness*, and how functional traits respond to urbanization. Spiders (Araneae) and carabids (Coleoptera, Carabidae) were selected as model groups, because of their well-defined ecology and taxonomy, together with their well-known response to urbanization, both in terms of taxonomic and functional diversity (e.g. Niemelä & Kotze, 2009; Sattler et al., 2010; Vergnes, Le Viol & Clergeau, 2012; Vergnes et al., 2014; Piano et al., 2017; Buchholz et al., 2018). The assemblages of the two selected groups were investigated by means of a hierarchical sampling design, combining data from isolated and connected control environmental patches along an urbanization gradient. The inner areas of traffic roundabouts were selected as isolated plots, while control patches were established on green areas connected with the surrounding environmental matrix. Specifically, we investigated whether: i) *nestedness* and *turnover* components differentially explain total variation (β-diversity) in taxonomic and functional diversity; ii) the urbanization level and patch isolation affect taxonomic and functional composition of carabid and spider communities, both in terms of homogenization (*nestedness*) and shift (*turnover*); and iii) functional traits show differential responses to the urbanization level and patch isolation.

Materials and methods

Sampling design

The study was carried out in 15 sampling plots (Fig. S1), randomly selected along an urbanization gradient in the municipality of Torino (about 880,000 inhabitants, NW-Italy). The average distance between the centre of sampling plots was 8,878 m (sd = ±1,705.6), ranging from 1,408 to 18,512 m. In order to test the effects of patch isolation on spiders and carabids, in each plot two sampling subplots were identified: i) one in an isolated patch, within a traffic roundabout; and ii) the other in the nearest green area, connected with the surrounding environmental matrix (connected control patch) (Fig. 1). Sampling plots consisted in circular areas of 150 m diameter (approximately 18,000 m^2), centred on the roundabout. Roundabouts ranged from 6 to 30 m radius (i.e. approximately from 150 to 2,500 m^2 of surface area), and we evaluated

111 their age using historical aerial maps on Google Earth (https://www.google.com/earth/download/ge/).
112 Based on the year of construction, we equally distributed them among three age classes (1 = less than 10
113 years; 2 = 10 years (built in 2007); 3 = more than 10 years) (see Appendix A: Table S1). Given that both
114 taxonomic groups were not affected by these factors, namely the area and the age of the roundabouts, we
115 did not include this information in our subsequent analyses. Connected control subplots were placed in the
116 closest green area within the sampling plot, being represented by small green urban patches (1,000 m^2),
117 urban parks or seminatural areas at the city borders (up to a few hundreds of hectares). The isolated patch
118 (roundabout) within a certain sampling plot was always smaller than the green area encompassing the
119 connected control subplot. In all sampling plots, subplots were located in comparable semi-natural
120 grassland habitats, maintained by regular mechanical mowing (see Fig. S2 for real examples). Roundabouts
121 were assumed to represent a good approximation of isolated patches, since they are completely
122 surrounded by roads, which likely constitute a selective barrier for our targeted arthropod groups.
123 The urbanization level was assessed in each sampling plot by extrapolating the impervious surface coverage
124 (ISA, Elvidge et al., 2007) from aerial images of the study area. We used the proportion (%) of impervious
125 surfaces as a proxy for the urbanization level, which was calculated from updated digital maps (year 2017)
126 by means of the QGIS software (QGIS Development Team, 2018) in a buffer of 1,600 m radius (see Fig. S1).
127 We tried as much as possible to avoid overlapping buffer areas. However, due to logistic constraints related
128 to permissions issued by the local authorities to access the areas of the roundabouts, our final design
129 implied an overlap of 9.6 km^2 between buffer areas, corresponding to about 8% of the total area covered
130 by the buffers (approximately 120 km^2). Given that carabid beetles and spiders often respond to factors at
131 different scales (Braaker et al., 2014; 2017), we checked whether this spatial scale represents the best
132 option to evaluate the response of ground arthropods to urbanization in the investigated area (see
133 Supplementary Materials).

134 *Data collection*

135 Ground-active arthropods were captured in each sampling subplot within a sampling section 30 m long
136 using three pitfall traps (5-10 m apart), placed at least 20 cm from the patch border, and at least 5 m apart

137 to increase catch efficiency (Ward, New & Yen, 2001). Plastic jars (diameter 90 mm, length 110 mm) were
138 dug into the ground so that the upper edge of the glass was flush with the soil surface, and filled with 20 ml
139 of 50% propylene glycol solution. All traps were emptied every three weeks, in three sampling sessions
140 between the 16th of May and the 18th of July 2017. The collected material was preserved in a 70% ethanol
141 solution. Spiders and carabids were sorted, identified and counted in the laboratory according to the keys
142 provided by Pesarini & Monzini (2010) and Boeken et al. (2002) for carabids, and by Nentwig et al. (2018)
143 for spiders. Data from pitfall traps collected in the same subplot were pooled together for subsequent
144 analyses.
145 It should be pointed out that pitfall traps provide data on activity density rather than real abundances of
146 species, thus overestimating extremely active species. Although the pitfall trap-based sampling method
147 introduces some bias in relative species abundances, the extent of the bias should be similar for each
148 sampling site.

149 *Functional traits*

150 Body length, dispersal capacity and trophic requirements – hunting mode for spiders and trophic level for
151 carabids – were considered as key functional traits, based on Buchholz et al. (2018).
152 Carabid species were assigned to two trophic groups (1 = zoophagous; 2 = phytophagous) according to
153 Vanbergen et al. (2010), and to three dispersal groups based on wing development (1 = brachypterous; 2 =
154 dimorphic; 3 = macropterous) according to Desender et al. (2008). The average body size was assigned to
155 each species according to Desender et al. (2008). Spider species were assigned to functional groups
156 reflecting their strategy of food provision, according to Cardoso et al. (2011). In order to obtain a
157 comparable number of species among categories, we referred to a broader classification (1 = hunters; 2 =
158 web-builders), without considering in detail the specific hunting strategies. Spiders were assigned to three
159 dispersal groups (1 = non- or sporadic ballooners; 2 = ballooners at juvenile stages only; 3 = ballooners at
160 juvenile and adult stages), based on Blandenier (2009), Bell et al. (2005) and Simonneau, Courtial and
161 Pétillon (2016). Each species' male and female average body sizes were extrapolated from Nentwig et al.

(2018). We retained only female's data due to the high correlation between the two measures (Pearson's correlation test: $r = 0.97$; $P < 0.001$).

Data analysis

All statistical analyses were performed using the *R* software (R Core Team, 2017).

Preliminary Analyses. In order to identify the best scale of response to the urbanization level for carabids and spiders, we calculated the degree of urbanization for each station in three progressive larger buffers, defined as circles with a 100 m (small scale), 400 m (medium scale) and 1600 m (large scale) radius - with the roundabout as the centre. For each taxonomic group, we proceeded with fitting three separated models, one for each scale of the urbanization level, after eliminating outliers in our dependent variables following the standard protocol for data exploration proposed by Zuur et al. (2009). We tested the response of the total abundance of carabids and spiders against the urbanization level, patch isolation and their interaction by means of Generalized Linear Mixed Models (GLMMs) (Zuur et al. 2009), performed with the function "glmer.nb" in the *lme4* package (Bates et al., 2015). We assumed a negative binomial distribution after checking for the overdispersion of our dependent variables. To account for the spatial and temporal dependency, a station (*PlotID*) and a session identifier (*Session*) were incorporated as random factors in the models. We selected the one with the lowest AICc and, given that the best response was observed at the large scale (see Appendix B: Table S2), our subsequent analyses were performed using the urbanization level measured in the buffer of 1,600 m of radius. We then tested whether the total abundance of carabids and spiders in the isolated patch is influenced by the area and the age of the roundabout and by the distance of the roundabout from the control patch with the same model structure. No corrections to the data from the isolated patches were introduced to keep into account the effects of the area and the age of the roundabout and the distance between the roundabout and the control patch, since both groups were not affected by these factors (see Appendix B: Table S3).

Multivariate statistics. Firstly, we investigated the variation in taxonomic and functional composition in spider and carabid assemblages by calculating taxonomic and functional β-diversity across the study area. β-diversity was here intended as dissimilarity among samples and it was calculated by means of the

188 complement of the Sørensen index, which ranges from 0 (samples are composed exactly by the same
189 species or functional groups) to 1 (samples do not share any species or functional groups). We used the
190 function “beta” in the *BAT* package (Cardoso, Rigal & Carvalho, 2015), which returns the overall β-diversity
191 and the contribution of its components, namely *turnover* and *nestedness*.
192 Secondly, we investigated the effects of the urbanization level and patch isolation on taxonomic and
193 functional composition of both groups with multivariate statistics, performed with the *vegan* package
194 (Oksanen et al., 2018). We first converted the urbanization level into a categorical variable by assigning
195 sampling plots to one of three categories of urbanization: low (ISA < 50%), intermediate (ISA = 50-70%) and
196 high (ISA > 70%). We then created a functional matrix, measuring the community-weighted means (CWM)
197 of trait values with the function “functcomp” of the *FD* package (Laliberté & Legendre 2010; Laliberté,
198 Legendre & Shipley, 2014). We built a site-by-trait matrix by multiplying the site-by-species matrix with a
199 species-by-trait matrix resulting in the CWM trait values, which represent the mean trait value of all species
200 in the community, weighted by their relative abundances, for body length (CWM-BL), dispersal (CWM-Disp)
201 and trophic composition (CWM-Trophic). Increasing values in CWM-Trophic indicates a shift towards
202 phytophagous species in carabids, and towards web-builders in spiders.
203 Taxonomic and functional composition among the three urbanization levels, and between control and
204 isolated patches, were visualized by means of a Principal Coordinate Analysis (PCoA). We then tested
205 whether the urbanization level and patch isolation were responsible for either taxonomic or functional
206 homogenization by means of the Test of Homogeneity for Multivariate Dispersion (Anderson, Ellingsen &
207 McArdle, 2006). This test measures the distance of each site to its associated group median calculated on a
208 site-by-site distance matrix and subjects these values to an ANOVA (9,999 permutations) to assess if the
209 variance differed among groups. For taxonomic diversity, the site-by-site distance matrix was computed on
210 the matrix of relative abundance of species using the Bray-Curtis distance, whereas the Gower distance was
211 applied to the CWM matrix to obtain the site-by-site distance matrix for functional diversity. To detect
212 possible shifts in taxonomic and functional composition among urbanization and isolation levels, we
213 performed a Permutational Multivariate Analysis of Variance (PERMANOVA; Anderson, 2001), specifying

214 urbanization, isolation and their interaction as factors. Statistical significance was tested via 9,999 random
215 permutations with the sampling session (*Session*) and the ID of each sampling plot (*PlotID*) as strata.
216 To identify which taxa are specifically linked to the urbanization level and patch isolation categories, an
217 Indicator Species Analysis was performed through the calculation of the *IndVal* (Indicator Value) index
218 (Dufrêne & Legendre, 1997), using the function "multipatt" in the *indicspecies* package (De Caceres &
219 Legendre, 2009).
220 Statistical models. The effects of the urbanization level, patch isolation and their interaction on community-
221 averaged functional traits were tested with Generalized Linear Mixed Models (GLMMs) (Zuur et al., 2009),
222 performed with the function "glmer" in the *lme4* package (Bates et al., 2015) assuming a normal error
223 distribution. To account for the spatial and temporal dependency, a plot (*PlotID*) and a session identifier
224 (*Session*) were incorporated as random factors in the models. The set of functional metrics employed in this
225 analysis as dependent variables were extracted from the previously calculated functional matrices. In
226 addition, in order to check for spatial autocorrelation, we calculated the observed Moran's *I* for each
227 model, comparing it with the expected values.
228
229 **Results**
230 A total of 215 out of 270 pitfall traps were retrieved during the whole sampling season. Overall, we
231 collected a total of 1,722 carabids, belonging to 52 species, and 4,811 spiders, belonging to 66 species, with
232 a mean of 5.28 ± sd = 3.49 (min = 4 and max = 13 species per subplot) and 6.86 ± sd = 3.57 (min = 1 and max
233 = 19 species per subplot) species and 21.8 ± sd = 22.4 (min = 1; max = 126) and 34.9 ± sd = 32.6 (min = 3;
234 max = 162) individuals per subplot respectively (see Appendix C: Tables S4-S5 for details on the recorded
235 species of carabids; and Appendix D: Tables S6-S7 for details on the recorded species of spiders).
236 The pairwise dissimilarity (Fig. 2) among carabid communities (total β-diversity) within the sampling area,
237 was on average higher for taxonomic (0.84 ± sd = 0.15) than for functional diversity (0.62 ± sd = 0.12). The
238 contribution of the *nestedness* component to total β-diversity was higher than the *turnover* component in
239 both taxonomic diversity (*turnover* = 0.34 ± sd = 0.26; *nestedness* = 0.51 ± sd = 0.28) and functional diversity

(*turnover* = 0.12 ± sd = 0.21; *nestedness* = 0.51 ± sd = 0.28). Similarly, total β-diversity was on average higher for taxonomic diversity (0.77 ± sd = 0.17) than for functional diversity (0.55 ± sd = 0.21) among spider communities within the sampling area and again the *nestedness* component showed a higher contribution than the *turnover* one for both taxonomic diversity (*turnover* = 0.32 ± sd = 0.24; *nestedness* = 0.44 ± sd = 0.27) and functional diversity (*turnover* = 0.10 ± sd = 0.10; *nestedness* = 0.44 ± sd = 0.27).

The test of homogeneity for multivariate dispersion showed significant functional homogenization of both groups along the urbanization gradient (Table 1 and Fig. 3), whereas significant taxonomic homogenization was observed in isolated patches and urbanized plots for spiders (Table 1 and Fig. 4). The PERMANOVA revealed significant shifts in taxonomic and functional composition along the urbanization gradient for both groups. Spider taxonomic and functional composition was affected also by patch isolation and the interaction between urbanization and isolation, whereas carabid taxonomic composition showed a shift from connected to isolated patches (Table 2).

The indicator species analysis highlighted no indicator species for urbanization categories in both groups. Some indicator species were identified for connected control subplots, namely *Harpalus serripes* (*IndVal* = 0.562; P = 0.001) and *Harpalus tardus* (*IndVal* = 0.385; P = 0.043) for carabids; *Trochosa ruricola* (*IndVal* = 0.549; P = 0.008), *Thanatus arenarius* (*IndVal* = 0.461; P = 0.036) and *Pardosa tenuipes* (*IndVal* = 0.366; P = 0.049) for spiders. No indicator species were identified for isolated subplots.

The response of carabid community-averaged functional traits revealed a significant increase of the proportion of phytophagous species with increasing urbanization level (range: 0.01-0.50), whereas no significant results were recorded for CWM-BL (body size) and CWM-Disp (Dispersal) (Table 3 and Fig. 5). Regarding spiders, body size was negatively affected by patch isolation, whereas the interaction term showed a significant positive effect, underlying that CWM-body size of spiders in isolated patches increases with increasing urbanization (Table 3). Both dispersal and trophic groups were significantly affected by all terms (Table 3). CWM-Disp (dispersal) showed an increasing trend along the urbanization gradient, ranging from 2.06 to 2.64, and higher values in the isolated (2.26 ± sd = 0.25) than in the control (2.16 ± sd = 0.34) subplots (Fig. 5), while the interaction factor displayed a negative effect. Regarding CWM-Trophic, the

266 relative proportion of web-builders significantly increased along the urbanization gradient (range = 0.27-
267 0.82) and it was significantly higher in isolated (0.68 ± sd = 0.27) than in control (0.69 ± sd = 0.30) subplots,
268 while a negative effect was observed for the interaction factor (Fig. 5).
269 Lastly, Moran's *I* revealed that CWM-Disp and CWM-Trophic, but not CWM-BL, of both carabids and spiders
270 show significant spatial autocorrelation (Table 4).
271
272 **Discussion**

273 In this work, we analyzed how urbanization and fragmentation drive the *turnover* and *nestedness* patterns
274 of taxonomic and functional diversity within two taxonomic groups of ground arthropods, namely carabids
275 and spiders. Since we consistently sampled the same habitat type (i.e. semi-natural grasslands) exposed to
276 the same management practices, we could clearly depict the effect of site location within the conurbation,
277 without confounding factors due to local habitat characteristics.
278 Our results showed that taxonomic variation was higher than functional variation within both carabid and
279 spider communities along the urbanization and isolation gradients. This is in accordance with the functional
280 redundancy concept (Lawton & Brown, 1993) stating that multiple species perform similar roles in
281 communities and ecosystems (e.g. Petchey et al., 2007). This would guarantee high ecological resilience to
282 disturbance, which is particularly crucial for the maintenance of ecosystem functions in extremely altered
283 habitats, like urban areas. Decomposing β-diversity into its components revealed that the same
284 mechanisms drive variation in taxonomic and functional composition in the two taxa studied. Since
285 *nestedness* was the dominant component, some sampling plots host a subset of species of other sites. We
286 can hypothesize that both carabid and spider communities are composed of both specialized species,
287 surviving only in few localities, and generalist species, which can exploit a broad range of ecological
288 conditions. In the same way, some functional traits are removed from some sites without being replaced,
289 with possible repercussions on ecosystem functionality.
290 Results of the test of homogeneity for multivariate dispersion showed that spiders were more affected by
291 the urbanization level and patch isolation than carabids. Taxonomic homogenization (i.e. *nestedness*) was

observed only in spiders and it was explained by both the urbanization level and patch isolation. In addition, the role of isolation as a driver of taxonomic homogenization was further confirmed by the indicator species analysis that revealed indicator species for connected control subplots only. According to these results, we may hypothesize that connected control subplots could provide source populations for the isolated ones, guaranteeing their survival in disturbed habitats, likely recalling the island biogeographic model in fragmented landscapes (MacArthur & Wilson, 1967; Leibold & Chase, 2017). However, no indicator species were identified for the urbanization categories. Indicator species may not have been detected because they are too scarce and irregularly distributed among the subplots within each category. This result may however suggest that the observed *nestedness* pattern along the urbanization gradient may be due to varying relative abundances, instead of an effective substitution of species.

Regarding functional composition, the urbanization level causes functional homogenization in both communities. This may be due mainly to a filtering process, supporting the hypothesis that urbanization alters the functional community composition by shifting species composition towards more functionally homogeneous assemblages. Conversely, patch isolation did not significantly affect the functional composition of both carabid and spider communities, underpinning how the urbanization level alone may explain functional variation among samples. This corroborates the hypothesis that stochastic events, like dispersal-related processes or ecological drift, determine species richness in isolated patches for both carabids and spiders (Chase & Myers, 2011; Leibold & Chase, 2017). These outcomes parallel Concepción et al. (2017) who found functional homogenization with increasing urbanization in vascular plants and birds. On the contrary, our results are in contrast with those reported by Brice, Pellerin and Poulin (2017), who observed both taxonomic and functional differentiation with increasing urbanization in plant communities. These contrasting results are likely due to the fact that plant diversity often increases in urban habitats because of the unique abiotic conditions and disturbance regimes, favoring exotic species (McKinney, 2008). In our case, we did not record any exotic species in the examined communities, thus, at least from this point of view, our results seem to reflect the filtering effect of urbanization without confounding factors. However, increasing differentiation might also emerge from differential human activities in urban

318 green spaces, facilitating certain species in some places or eliminating others elsewhere. This could have
319 also happened in our case with spiders and carabid beetles, which could be easily transported in the soil or
320 in plant pots as eggs, juvenile stages or even as adults.
321 The results of the PERMANOVA performed on taxonomic composition pointed out a combined effect of the
322 urbanization level and patch isolation on the taxonomic shift—i.e. *turnover* component in both taxonomic
323 groups. The role of urbanization as a driver of this process has already been pointed out by Knop (2016) for
324 canopy insects, who observed true species replacement in terms of relative abundances in three insect
325 groups, including ground beetles, when comparing rural against urban areas. Regarding functional
326 composition, urbanization caused a shift in both examined groups, whereas patch isolation affected only
327 spiders, and these trends are confirmed by the analysis of the response of functional traits.
328 Dispersal capacity significantly increased along the urbanization gradient for spiders. In particular, this
329 increasing trend was clear in connected control subplots, while isolated subplots hosted highly dispersive
330 species, regardless of the urbanization level. High dispersal capacity is essential in disturbed habitats in
331 order to escape adverse environmental conditions and to enhance population survival in the area; vice-
332 versa in stable, late-successional habitats more energy can be invested in reproduction, favoring the
333 establishment of less dispersive species (Roff, 1975). Hence, the observed decline in low-dispersive species
334 appears in accordance with the dynamic nature of urban environments, where human activities result in
335 high turnover rates of suitable habitat patches (Parris, 2016). The observed high dispersal capacity of
336 spiders in isolated subplots can be explained in the framework of the theory of island biogeography
337 (MacArthur & Wilson, 1967), which predicts that isolation of suitable patches increases extinction rates.
338 Therefore, only highly dispersive species can support viable populations in isolated patches, due to the
339 continuous immigration of new individuals, increasing the average dispersal capacity of the community.
340 Surprisingly, we did not record any effect of the urbanization level or patch isolation on carabid dispersal
341 capacity, in contrast with the available literature (Piano et al., 2017). The differential response obtained for
342 the two examined arthropod groups might be due to their different dispersal mode, i.e. active dispersal in
343 carabids and passive dispersal in spiders. The peculiar dispersal mode of spiders, i.e. ballooning, requires

particular physical conditions to occur (Weyman, 1993; Simonneau, Courtial & Pétillon, 2016) and dispersing individuals have no control over the flight direction (Compton, 2002). In heavily fragmented landscapes, the probability to reach isolated patches is therefore strictly related to the ballooning performance, which is maximized in highly dispersive species (Bonte et al., 2003). Conversely, carabid dispersal, either cursorial or by flight, is active and not constrained by the physical environment, therefore all species likely have the same probability to reach isolated patches.

The analysis of the trophic structure revealed how urbanization strongly acts on both examined taxa. Regarding carabids, we observed a significant replacement of zoophagous with phytophagous species. Phytophagous carabids are specialized on seeds from ruderal plants (Thiele, 1977; Honek et al. 2007; Honek, Martinkova & Saska, 2011), which typically occur in highly impermanent sites (Ribera et al., 2001). Phytophagous species can therefore be considered adapted to ruderal habitats (Vanbergen et al., 2010), likely supporting viable populations within urban sites in our study. Concerning spiders, the proportion of web-builders significantly increases with increasing the urbanization level in connected control subplots. Conversely, in isolated subplots, web-builder proportion is higher than connected control subplots regardless of the urbanization level, suggesting how the effect of patch isolation overrides urbanization in determining the trophic structure of spiders. This may be due to the fact that patch isolation may negatively affect preys of cursorial spiders (Hawn et al., 2018) with consequent negative effects on hunters (Gravel et al., 2011; Zalewski et al., 2018). However, it should be pointed out that this shift from hunters to web-builders in isolated patches might also be a consequence of the dispersal-based selection, since, in our study, web-builders include most of the highly dispersive species (i.e. linyphiids).

Contrary to our expectations, body size did not respond to the urbanization gradient in our study for both taxonomic groups, in accordance with Buchholtz et al. (2018). However, we observed a significant decrease of spider body size along the urbanization gradient in connected control patches, but not in isolated ones. It has been demonstrated that community-wide body-size shifts occur in urban communities as a consequence of increased temperatures due to the urban heat-island effect, but these shifts are mediated by the dispersal capacity of each taxon (Merckx et al., 2018). Thus, this lack of response may result from the

dispersal-mediated effect, since higher dispersal capacity of spider species recorded in roundabouts may dampen the urban heat-island effect on body size.

Results of the Moran's *I* test revealed that there is high spatial autocorrelation among sampling plots in terms of dispersal capacity and trophic composition, but not of body size, for both groups. These patterns underlie the role of the spatial arrangement of individuals in a landscape, which is rarely random (McGlinn et al., 2019). Instead, most individuals are spatially clustered or aggregated in some way, with repercussions also on the functional traits, as demonstrated by our results.

Conclusions

Overall, we here highlighted differential mechanisms underlying the selective pressure exerted by urbanization and fragmentation on ground arthropods, showing a more evident response in spiders than carabids. This is in accordance with literature, which highlighted how top predators are usually more sensitive to urbanization than lower trophic levels (Egerer et al., 2017; El-Sabawii, 2018).

We have demonstrated that *nestedness* more than *turnover* explains taxonomic and functional variation in the examined communities, indicating a loss of species and functionality among sampling plots. This homogenization process is mainly due to urbanization, which filters species based on their functional traits in both the examined groups.

In addition, the high spatial autocorrelation among sampling plots identified for dispersal capacity and trophic composition underlies that stochastic factors, e.g. source-sink dynamics, also play a role in driving the functional composition of examined communities. Stochastic events, like those caused by human activities, may cause the extinction of some species in the examined sampling plots, potentially compromising the species survival in the whole sampling area. This may have repercussions on ecosystem functionality if key stone species are removed from the landscape.

Our approach, thus, proved to be particularly useful to understand how different facets of urbanization affect biodiversity, providing a valuable framework to predict how biotic communities will respond to increasing anthropogenic pressures associated with urbanization. In the next future, further investigations

396 should be performed to highlight alterations at the ecosystem level in order to provide management
397 suggestions aiming at reducing negative effects caused by urbanization.
398
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407
408 **Appendix A. Supplementary data**
409 **Appendix B. Supplementary data**
410 **Appendix C. Supplementary data**
411 **Appendix D. Supplementary data**
412 **Supplementary data associated with this article can be found, in the online version, at XXXXX**

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602 **FIGURE CAPTIONS**

603 Fig. 1. Schematic representation of the nested sampling design adopted in this research. Sampling plots
604 were placed along an urbanization gradient, each one being composed of two subplots (I = isolated; C =
605 connected control). In each sub-plot, three pitfall traps were activated during the sampling period (black
606 dots).

607 Fig. 2. Barplot representing the contribution of *turnover* (dark grey) and *nestedness* (light grey) to the total
608 taxonomic and functional β-diversity for carabids (left panel) and spiders (right panel).

609 Fig. 3. Ordination of carabid (left) and spider (right) communities, according to the first two PCoA axes
610 performed on taxonomic (upper panel) and functional (lower panel) composition. Ellipses represent
611 standard deviations around the medians of urbanization levels (green = low; orange = intermediate; purple
612 = high). Different symbols represent sampling subplots belonging to different urbanization levels (circles =
613 low; squares = intermediate; triangles = high). The animal silhouettes are from PhyloPic
614 (http://www.phylopic.org).

615 Fig. 4. Ordination of carabid (left) and spider (right) communities, according to the first two PCoA axes
616 performed on taxonomic (upper panel) and functional (lower panel) composition. Ellipses represent
617 standard deviations around the medians of isolation levels (blue = connected control; brown = isolated).
618 Different symbols represent sampling subplots belonging to different isolation categories (circles =
619 connected control; squares = isolated). The animal silhouettes are from PhyloPic
620 (http://www.phylopic.org).

621 Fig. 5. Predicted CWM values (CWM-Trophic = trophic composition; CWM-BL = body length; CWM-Disp =
622 dispersal) and confidence intervals along the urbanization gradient for carabids and spiders (light blue line =
623 connected control patch data; orange line = isolated patch data). Only significant trends are reported. The
624 animal silhouettes are from PhyloPic (http://www.phylopic.org).

Figure 1

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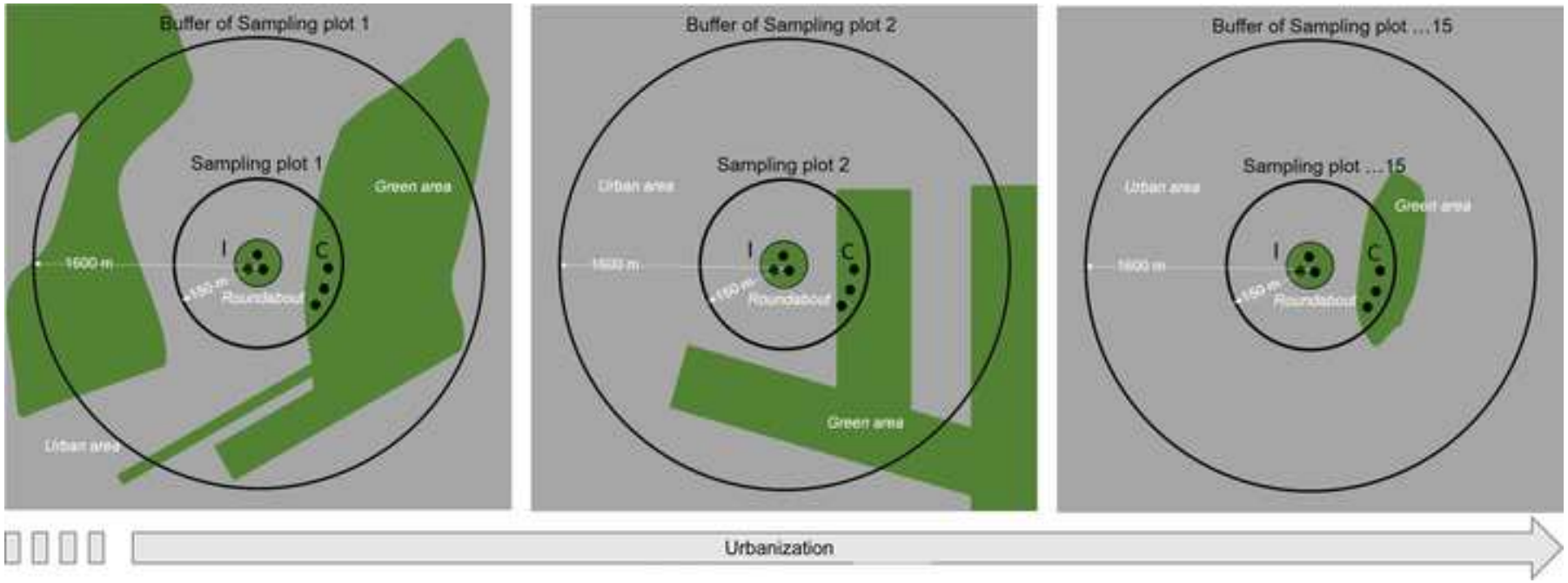

Figure 5

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